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5.1 OVERVIEW

Diazinon is released to the environment solely by human activities. Major atmospheric emissions result from volatilization of the chemical from soil resulting from its use as a soil nematocide and insecticide or from drift during pesticide application. Diazinon is released to surface waters directly by point source discharges, from drift during pesticide applications, and by runoff from agricultural and urban areas. No current information is available on total environmental releases of diazinon from production and processing facilities to air, water, and soil because these facilities were not required to report releases to the Toxics Release Inventory until January 1, 1995 (EPA 1995a, 1995b).

Diazinon is found in all environmental compartments, but shows no pronounced tendency to partition to a particular environmental medium. Given adequate time, diazinon will be degraded by abiotic and biotic processes so that the parent compound is not persistent. Diazinon has been detected in the atmosphere and trace amounts of its oxygen analogue (diazoxon) have also been detected. The oxon to thion ratio ranged from 0.056 to 7.1, but was generally less than 0.4 (Glotfelty et al. 1990a). In a study of diazinon use in the Central Valley of California, Seiber et al. (1993) reported that during daylight hours, the oxon to thion ratio in the atmosphere averaged 0.52, while at night the ratio was 0.10. Diazinon can be converted to diazoxon in the atmosphere via ultraviolet radiation (UV) (Aizawa 1989). The estimated half-life for the vapor phase reaction of diazinon with hydroxyl radicals is approximately 4 hours (SRC 1995). Diazinon can be transported moderate distances in the air from its original point of use (Zabik and Seiber 1993).

Diazinon released to surface waters or soil is subject to volatilization, photolysis, hydrolysis, and biodegradation. Biodegradation, primarily under aerobic conditions, is a major fate process for diazinon associated with water and soil. Diazinon can be biodegraded under anaerobic conditions as well. Hydrolysis is an important mechanism for degradation, particularly at low pH in-water and soil. Diazinon has a relatively short half-life in water, ranging from 70 hours to 12 weeks depending on pH, temperature, and sunlight as well as the presence of microorganisms (Chapman and Cole 1982; Ferrando et al. 1992; Frank et al. 1991b; Scheunert et al. 1993; Schoen and Winterlin 1987; Sharom et al. 1980b; Wolfe et al. 1976). The half-life of diazinon in soil is influenced by the pH conditions in the soil and the soil type. The half-life values at pH 4, 7, and 10 were 66, 209, and 153 days,

respectively, in sandy loam; 49, 124, and 90 days, respectively, in clay loam; and 14, 45, and 64 days, respectively, in sandy loam amended with peat (Schoen and Winterlin 1987). Diazinon is moderately mobile in some soils, particularly those with an organic matter content less than 3%, and can leach from soil into groundwater. If released to water, this pesticide does not bioaccumulate (BCF values generally less than 100) in aquatic organisms.

In the United States, monitoring efforts under many national programs have not analyzed for this chemical. In addition, environmental monitoring efforts in general have decreased noticeably within the last decade. This makes it difficult to provide current quantitative estimates on the fate and transport of diazinon in various environmental compartments. Diazinon has been identified in air samples from both rural and urban areas and in indoor air in both domestic and commercial buildings. It has also been detected in surface water, effluents from publicly owned treatment works (POTWs), and groundwater. It has been detected in soil and sediment in areas where it is extensively used in agriculture. Current information is lacking on the total amount of diazinon released to the environment and on the amount of diazinon that partitions into each environmental compartment.

The best-documented concern over diazinon relates to acute exposures of humans during or immediately following pesticide applications. This concern is warranted, since diazinon is still widely used, with many applications in urban areas (homes and gardens) that increase the possibilities of human exposure. Diazinon and its major metabolite, diazoxon, have significant acute toxicity to humans. The predominant exposure pathway for the general population appears to be via dermal exposure during application of the compound for commercial and domestic pest control uses and via inhalation in indoor air spaces during or immediately following application. Dermal exposure is probably a more significant exposure of people using the pesticide for home and garden applications. Exposure to diazinon can also result from ingestion of contaminated food and water. Exposure is greatest for those individuals occupationally exposed to diazinon, particularly those involved in its production and manufacture; those involved in its application for agricultural, commercial, or domestic pest control uses; and those involved in its disposal at hazardous waste sites.

Diazinon has been identified in at least 18 of the 1,430 current or former EPA National Priorities List (NPL) hazardous waste sites (HazDat 1996). However, the number of sites evaluated for diazinon is not known. The frequency of these sites within the United States can be seen in Figure 5-l.

Figure 5-1. Frequency of Sites with Diazinon Contamination

SITES

[ ] 1

[ ] 2-3

[ ] 4-5

[ ] 6-7
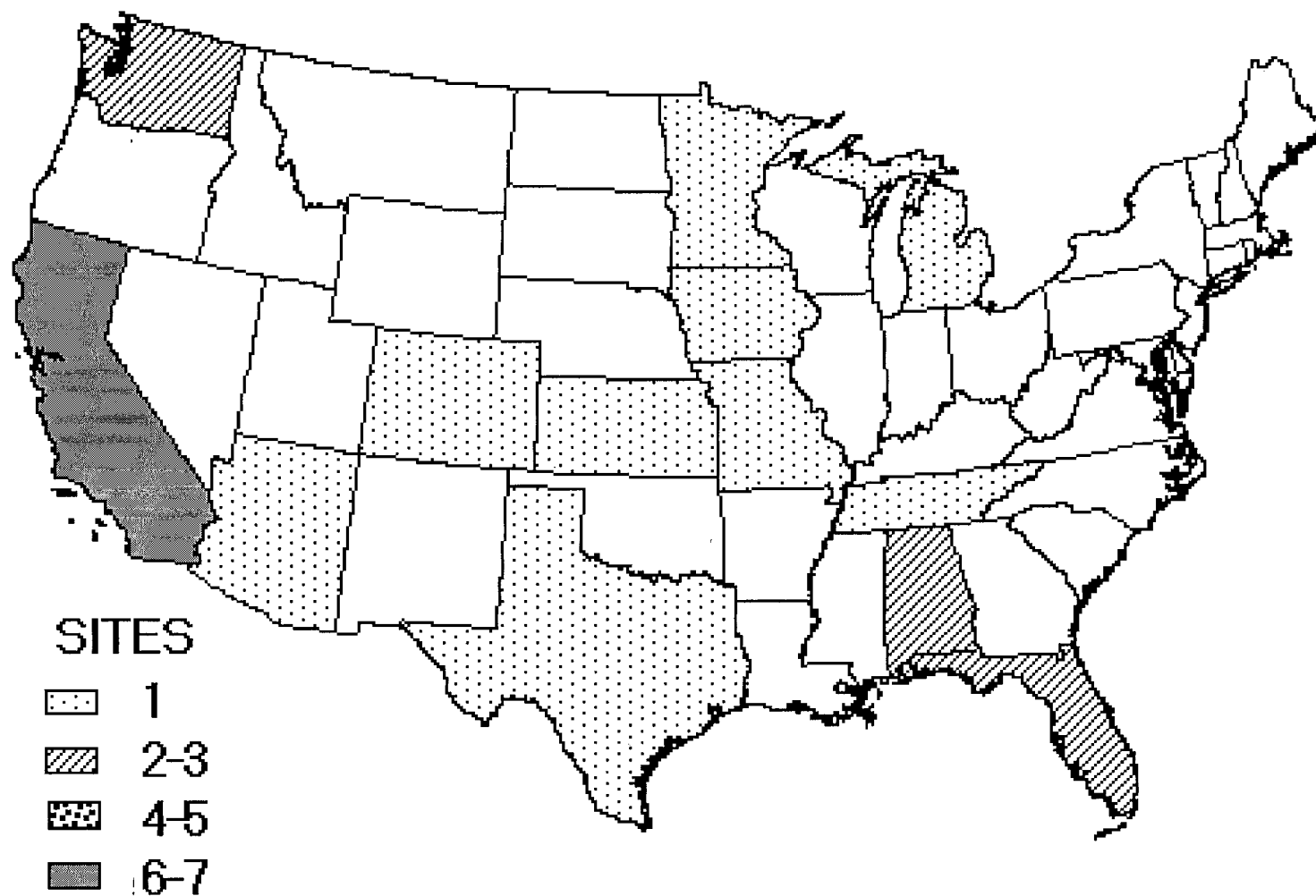
Derived from HazDat 1996

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5.2 RELEASES TO THE ENVIRONMENT

Prior to January 1, 1995, facilities involved in the production or processing of diazinon were not required to report the amount of releases to various environmental matrices (EPA 1995a, 1995b). Data on diazinon releases for 1995 will be available in the Toxics Release Inventory database in 1997.

Diazinon has been identified in a variety of environmental media (surface water, leachate, groundwater, soil and sediment) collected at at least 18 of the 1,430 current or former EPA National Priorities (NPL) hazardous waste sites where diazinon was detected in some environmental media (HazDat 1996).

5.2.1 Air

Diazinon is released into the atmosphere solely by human activities associated with its production and use as a pesticide. These releases include releases to ambient air from production, and from agricultural or domestic lawn and garden applications, and releases to indoor air from pest-control treatment of domestic and commercial buildings. It appears that diazinon that has been applied to a field can undergo volatilization to the atmosphere (Glotfelty et al. 1990a; Schomburg et al. 1991; Seiber et al. 1993; Zabik and Seiber 1993). Glotfelty et al. (1990b) estimated that up to 24% of the diazinon applied to dormant peach orchards may be released through long-term volatilization losses even though volatilization quickly declines to low levels. Volatilization from the home and garden applications that now may account for over 40% of total diazinon usage is impossible to estimate.

No information was available in the Toxics Release Inventory database on releases of diazinon to air from manufacturing and processing facilities because these facilities were not required to report releases of this chemical prior to January 1, 1995 (EPA 1995a, 1995b).

Diazinon has not been identified in air samples collected at any of the 18 current or former NPL hazardous waste sites where it was detected in some environmental media (HazDat 1996).

5.2.2 Water

Diazinon is released into water directly from point source discharges, from drift during pesticide applications, and from nonpoint source runoff from agricultural and urban areas. Since diazinon is not

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a Priority Pollutant under the Clean Water Act, it has not been evaluated extensively in water quality monitoring programs in rivers, lakes, or estuaries. Recently the use of permit compliance bioassay testing has helped identify point source discharges with acutely toxic effluents, and follow-up chemical analyses have pinpointed the identity of specific toxicants (Amato et al. 1992). Such work has led to the identification of diazinon as a cause of toxicity in POTW discharges (Amato et al. 1992; Burkhard and Jenson 1993). This is not surprising given the widespread use of diazinon in urban areas to control indoor pests and lawn and garden pests. It is easy for diazinon and its residues to reach the sewer collection systems for many POTWs.

In addition to loadings passing through sewage treatment systems, diazinon can reach surface waters directly from point source discharges (Braun and Frank 1980), from nonpoint source inputs introduced from agricultural (Braun and Frank 1980; Kendall et al. 1993; Kuivila 1993; Maguire and Tkacz 1993; Szeto et al. 1990; Wan et al. 1994), or from suburban runoff (Frank et al. 1991b). It is impossible to obtain estimates of these loadings to surface waters. Water concentrations and transport of diazinon through the Sacramento-San Joaquin Delta and the adjacent portions of San Francisco Bay were studied in 1993 by the U.S. Geological Survey (Kuivila 1993). Diazinon was applied as a dormant spray in the Central Valley of California during 2 weeks of dry weather in January 1993. Pulses of elevated diazinon concentrations were detected in the Sacramento and San Joaquin Rivers after a series of rainstorms in early February 1993. All concentrations of diazinon measured in river and bay water samples exceeded 9 ng/L (9 ppt). Contaminated water samples collected from the San Joaquin River produced 100% mortality in bioassay tests conducted with Cerioduphnia dubia for 12 consecutive days from February 8 to 19. The mortality of this sensitive indicator species was attributed to agricultural runoff of diazinon associated with the February rain events (Kuivila 1993).

No information was available in the Toxics Release Inventory database on releases of diazinon to water from manufacturing or processing facilities because these facilities were not required to report releases of this chemical prior to January 1, 1995 (EPA 1995a, 1995b).

Since diazinon is moderately mobile in soils under certain conditions, it has the potential to migrate through the soil and into groundwater. Detections have been made in some groundwater wells in the United States (Cohen 1986; EPA 1989a). In areas with heavy applications of diazinon combined with irrigation or water-level adjustment techniques, diazinon detections in groundwater also have been

documented (Cohen 1986; Frank et al. 1987, 1990b). It has not been possible to obtain quantifiable estimates of these diazinon loadings to groundwater.

Diazinon has been identified in surface water, leachate, and groundwater samples collected at 4, 1, and 9 of the current or former NPL hazardous waste sites, respectively, where it was detected in some environmental media (HazDat 1996).

5.2.3 Soil

Diazinon is released into soils primarily from its registered use on various agricultural crops and its use in home garden and lawn applications. Soils are the target for the vast majority of diazinon applications both as a nematocide and as an insecticide agent. In agricultural areas, diazinon may also be transferred to aquatic sediments (Domagalski and Kuivula 1993; Szeto et al. 1990; Wan et al. 1994). Since diazinon undergoes various activation and degradation reactions in the course of time ranging from hours to months, these loadings to soils and sediments are a temporary phenomena. The absence of current statistics on pesticide production, sales, and/or usage make it difficult to estimate releases to soils or sediments.

No information was available in the Toxics Release Inventory database on releases of diazinon to soil from manufacturing and processing facilities because these facilities were not required to report releases of this chemical prior to January 1,1995 (EPA 1995a, 1995b).

Diazinon has been identified in top soil samples (<3 inches deep), subsurface samples (>3 inches deep), soil samples with unspecified depth, and in sediment samples collected at 4, 2, 4, and 4 current or former NPL hazardous waste sites, respectively, where diazinon was detected in some environmental media (HazDat 1996).

5.3 ENVIRONMENTAL FATE

Diazinon can move into various environmental compartments, but there does not appear to be a major reservoir or sink for this chemical in any specific environmental compartment primarily because of its relatively rapid degradation in each environmental medium.

5.3.1 Transport and Partitioning

Based on its vapor pressure (see Table 3-2), if diazinon is released to the atmosphere, it will be expected to exist both in the vapor phase and particulate phase (Eisenreich et al. 1981). Glotfelty et al. (1990a) reported that during stagnant inversion fog events in the Central Valley of California, 56% and 19% of the diazinon in the air-phase was associated with vapor and aerosol particles, respectively, and only 24% of the diazinon was dissolved in the water phase. Schomburg et al. (199 1) reported slightly different distributions for fog events resulting from advected oceanic fog. In this study, 26% and 10% of the diazinon in the air-phase was associated with vapor and aerosol particles, respectively; 62% of the diazinon was dissolved in the water phase. Zabik and Seiber (1993) studied the atmospheric transport of diazinon from California's Central Valley to the Sierra Nevada Mountains. These samples collected during January through February 1991 represented the simultaneous collection of both vapor and particulate phases. Concentrations of diazinon and its oxon ranged from 13 to 10,000 pg/m^3 and 4 to 3,000 pg/m^3, respectively, for samples collected at the 114 m elevation and from 1.4 to 12 pg/m^3 and 1.8 to 13 pg/m^3, respectively, at the 533 m elevation. The pesticide concentrations in air samples decreased with distance and elevation moving east from the Central Valley into the higher elevations of the Sierra Nevada Mountains. At times, air concentrations at the 114 m elevation were 1,000 times greater than concentrations detected at 533 m elevation. Concentrations at the 1,920 m elevation were typically below the limit of quantification. Wet deposition samples collected at the 114 m elevation contained up to 6,100 pg/mL diazinon and 2,300 pg/mL diazinon oxon.

Limited data based on atmospheric sampling and laboratory studies (Glotfelty et al. 1990a, 1990b) suggest a much greater potential for diazinon transport into the atmosphere after application to soils and vegetation. While the activation process (thiono to oxon conversion) in the air would tend to transform diazinon fairly rapidly, the possibility of atmospheric transport means that this pesticide can move some distance from agricultural to nonagricultural areas (Glotfelty et al. 1990a, 1990b; Schomberg et al. 1991; Seiber et al. 1993; Zabik and Seiber 1993).

Diazinon released to water from both point and nonpoint sources may be emitted to the atmosphere by volatilization, sorbed to soils and sediments, or accumulated in aquatic organisms. While evaporation may not be expected to be significant based upon the Henry's law constant (see Table 3-2), volatilization of diazinon can be an important transport process. Sanders and Seiber (1983) reported

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that 17% of the diazinon added to a model pond volatilized in 24 hours. Diazinon released to water also may be adsorbed moderately by soils and sediments based on its organic carbon partition coefficient (K_{oc}) values measured in soil (Sharom et al. 1980a). Because this pesticide is only moderately adsorbed by some soils, leaching into groundwater can occur.

Diazinon does not significantly bioaccumulate in aquatic organisms. A comparison of biological concentrations factor (BCF) values obtained for various freshwater and saltwater fish and invertebrate species is presented in Table 5-l. The BCF values generally range from 4 to 337, but there are only a few cases where the measured BCF value for diazinon exceeds 100. In those experiments where testing was continued for several days after exposure to the diazinon had ended, tissue residues generally decreased rapidly within l-5 days (El Arab et al. 1990; Sancho et al. 1993; Tsuda et al. 1989, 1990, 1995). Despite the fairly low BCF values, some researchers still recommend caution in consuming some aquatic species (EPA 1993; Keizer et al. 1991). This is in large measure because the mechanisms that fish and invertebrates use to metabolize diazinon are poorly understood and seem to vary widely from species to species. In addition, diazinon and its metabolites have not been widely monitored in aquatic species. Since some of the metabolites of diazinon are themselves toxic, a measure of caution may still be in order in cases where there is reason to believe edible fish or shellfish have had recent exposure to diazinon (Keizer et al. 1991). This is partially the basis for the EPA recommendation to states to consider routine monitoring for diazinon in edible fish and shellfish species as part of their state toxics monitoring programs particularly in those watersheds where extensive use of diazinon is identified (EPA 1993).

Diazinon released in soil from its registered uses partitions to the atmosphere through volatilization, to surface water via runoff, and to groundwater as a result of leaching. According to Kenaga (1980), chemical compounds with an K_{oc}, of <100 are considered moderately to highly mobile; diazinon with a K_{oc} value of 40-432 (mean of 191), therefore, would be considered moderately mobile. Additional parameters influencing the leaching potential of this chemical include the soil type (e.g., clay versus sand), the amount of rainfall, the depth of the groundwater, and the extent of degradation. In laboratory tests of sand and organic soil, Sharom et al. (1980a) found that 26, 22, 11, 11, and 7% of the diazinon leached from sand (after 5 successive 200 mL rinses), respectively. A total of 95% of the diazinon added to the sand leached after 10 successive 200 mL rinses. In organic soil, however, only 3, 4, 11, 9, and 7% of the diazinon leached from soil (after 5 successive 200 mL rinses), respectively. Only 50% of diazinon added to the organic soil leached after 10 successive 200 mL rinses. While

Table 5-1. Bioconcentration Data for Diazinon

Species common name Scientific name	Exposure type	Duration (days)	BCF[a]	Reference
Freshwater				
Shrimp *Paratya compressa compressa*	F	3	4	Seguchi and Asaka 1981
Oriental weatherfish *Misgurnus anguilli candatus*	F	14	28	Seguchi and Asaka 1981
Common carp *Cyprinus carpio*	F	3	130	Seguchi and Asaka 1981
Perch *Sarotherodon galilaeus*	S	3	39	El Arab et al. 1990
Rainbow trout *Oncorhynchus mykiss*	F	3	92	Seguchi and Asaka 1981
Brook trout *Salvelinus fontinalis*	F	210	25	EPA 1977
Guppy *Poecilia reticulata*	R	2	39	Keizer et al. 1991
Zebra fish *Brachydanio rerio*	R	2	300	Keizer et al. 1991
Willow Shiner *Gnathopogon caerulescens*	F	7	248	Tsuda et al. 1989
Killifish *Oryzias latipes*	F	3	20	Tsuda et al. 1995
Fathead minnow *Pimephales promelas*	F	2–304	337[b]	Veith and Kosian 1983
Saltwater				
Sheepshead minnow *Cyprinodon variegatus*	F	108	147	Goodman et al. 1979

[a]BCF listed is the highest BCF value reported in the cited reference.
[b]Calculated quantitative structure-activity relationship (QSAR) value as reported in ASTER.

F = flow-through exposure system; S = static system; R = renewal system

diazinon can show sorption in soils with high organic content (>3%), in most other soil types diazinon has properties suggesting a moderate potential for leaching into groundwater (Arienzo et al. 1994; Sharom et al. 1980a). Arienzo et al. (1994) tested the adsorption and mobility of diazinon in 25 soils with different physicochemical properties. Diazinon was found to be slightly mobile in 80% of the soils tested (those with organic matter content <3%), and immobile in 20% of the soils tested (those with organic matter content >3%). The compound leached primarily from light soils with low organic matter content. Levanon et al. (1994) assessed the impact of plow tillage on microbial activity and the fate of diazinon and other pesticides in the top 5 cm soil layer. A higher leaching rate for diazinon was detected in plow tillage soils than in no-tillage soils after incubation for 21 days. The no-tillage soils were characterized by a higher organic matter content and higher microbial populations and activity than the plow tillage soils.

Arienzo et al. (1993) conducted a study of adsorption and mobility of diazinon in soils from aqueous media and mixtures of methanol-water and hexane-water. Adsorption of diazinon by soils from aqueous systems was related to organic matter content (i.e., the higher the organic content, the greater the adsorption). In methanol-water and hexane-water systems, the adsorption of diazinon by soils decreased. This situation may arise at hazardous waste disposal sites where pesticide waste residues and cosolvents may be encountered together. The presence of these organic solvents will increase the mobility (leachability) of diazinon in the soil and increase the potential for groundwater contamination. Diazinon has been detected in groundwater in the United States (Cohen 1986; EPA 1989a; HazDat 1996), and in the Great Lakes region of Ontario, Canada (Frank et al. 1987, 1990b).

5.3.2 Transformation and Degradation

Diazinon is subject to a variety of abiotic and biotic degradation processes in all environmental compartments.

5.3.2.1 Air

Diazinon, once released to the atmosphere, may be subject to direct photolysis since it absorbs light in the spectra above 290 nm (Gore et al. 1971). Glotfelty et al. (1990a), Schomburg et al. (1991), Seiber et al. (1993), and Zabik and Seiber (1993), all reported the presence of diazinon and its activated product (diazoxon) in atmospheric samples. Glotfelty et al. (1990a) believe that diazoxon is formed

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by atmospheric oxidation especially during the daylight hours. Schomburg et al. (1991) reported that diazinon undergoes transformation to diazoxon during atmospheric transport from agricultural to nonagricultural areas. Seiber et al. (1993) reported mean concentrations of diazinon of 76.8 ng/m^3 and of diazoxon of 10.8 ng/m^3 in air samples collected near fruit and nut orchards in Parlier, California. The half-life (first-order kinetics) for the vapor phase reaction of diazinon with hydroxyl radicals in the atmosphere is estimated to be 4 hours, assuming an atmosphere containing 5×10^5 hydroxyl radicals/m^3 at 25 °C (SRC 1995).

5.3.2.2 Water

Diazinon released to water may be subject to both abiotic degradation (i.e., hydrolysis and photolysis) and biotic degradation by microorganisms. The rate of abiotic degradation is influenced strongly by pH and temperature. In a laboratory study, Chapman and Cole (1982) reported that pH alone influenced the half-life of diazinon maintained in sterile water-ethanol (99:l) phosphate buffer solutions at 25 °C. Degradation of diazinon was most rapid under acidic conditions with half-life values in weeks (days shown in parentheses) (first-order kinetics) of 0.45 (3.15), 2.0 (14), 7.8 (54.6), 10.0 (70), and 7.7 (53.9) at pH values of 4.5, 5.0, 6.0, 7.0, and 8.0, respectively. Garcia-Repetto et al. (1994) also studied the influence of pH on the degradation of diazinon in water-ethanol (9: 1) solutions maintained between 15 and 31 °C. These authors reported estimated half-life values (first-order kinetics) for diazinon of 1.31, 8.57, and 8.19 days at pH values of 2, 7.5, and 8.7, respectively. The higher temperatures and lower pH conditions of this study may account for the more rapid degradation rates. Frank et al. (1991b) followed the degradation of diazinon in natural surface/groundwater samples at pH 8.2 that were either stored in the laboratory at 4 °C in the dark or at 21 °C under ambient indoor fluorescent light conditions for 125 days. Under the two temperature and light regimes the half-life values (first-order kinetics) of diazinon were 14 days (light at 21 °C) and 45 days (dark at 4 $^\circ$C). Degradation was more affected by temperature suggesting that hydrolysis was the primary mode of degradation.

Wolfe et al. (1976) reported that diazinon absorbs sunlight less than some of its organophosphate relatives, but that diazinon undergoes direct photolysis in water. The estimated half-life (first-order kinetics) for photolysis in aqueous solutions maintained in glass cells and irradiated with a mercury vapor lamp (>290 nm) was 1,000 hours (42 days). Frank et al. (1991b) investigated the degradation of diazinon in surface and groundwater samples, but found little difference in the rate of diazinon

degradation in light and dark conditions. The half-life (first-order kinetics) of diazinon of 88 days
(light) and 99 days (dark) suggests that photolysis was not a major factor in degradation.

Scheunert et al. (1993) studied the effects of photodegradation (via exposure to sunlight) on diazinon
dissolved in distilled water, in a humic acid aqueous solution, and in natural water samples from the
Isar and Rhine Rivers and Lake Ammersee in Germany with comparable samples maintained in the
dark at 25 °C. In the dark, river water had a higher diazinon degradation capacity than distilled water.
This was attributed to the oxygen and hydroxyl ion concentration of the river water. The degradation
capacity of natural water samples was further enhanced by exposure to sunlight. The highest
degradation capacity was observed for the Rhine River water which also had the highest oxygen and
hydroxyl ion concentration and the highest pH value (8.1) of the natural waters tested.

Sharom et al. (1980b) studied the degradation of diazinon under laboratory conditions using both
distilled water and natural water samples. Degradation was more rapid in natural water (pH 7.7)
(12 weeks) than in sterilized natural water, sterilized distilled water, or distilled water (>16 weeks),
suggesting that biodegradation of diazinon was occurring. Ferrando et al. (1992) conducted a
laboratory microcosm study using both natural surface water and tap water. These experiments were
conducted in aerated aquaria, maintained at 22 °C with a 12-hour 1ight:dark period. The pH of the
natural water was 9.0 and that of the tap water was 7.5. The half-life values (first-order kinetics) of
71 hours and 79 hours for the natural and tap water samples, respectively, both indicate rapid
degradation. Under these experimental microcosm conditions, hydrolysis, photolysis, and
biodegradation may all be operative in the natural water system. Wide discrepancies in the rates of
diazinon degradation in water reported in the literature appear to be influenced by both abiotic and
biotic factors.

Although diazinon has been detected in groundwater samples in both the United States and Canada
(Cohen 1986; EPA 1989a; Frank et al. 1987, 1990b; HazDat 1996), no studies were identified
concerning diazinon- transformation and degradation processes within aquifers. Based on-theoretical
considerations, abiotic hydrolysis mechanisms would be expected to degrade diazinon within a few
months (Chapman and Cole 1982; Cowart et al. 1971).

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5.3.2.3 Soil Transformation and Degradation

Once released to soils and sediments, diazinon can be degraded by hydrolysis, photolysis, and biodegradation by several genera of microorganisms. Microbial degradation appears to be the major pathway for the degradation of diazinon in soils; however, under anaerobic conditions, abiotic hydrolysis appears to be the most probable mechanism responsible for degradation of the compound under acidic soil conditions (EPA 1990b).

The influence of soil pH on the persistence of diazinon was studied by Chapman and Cole (1982). Diazinon degradation was found to be more rapid in organic soils with pH values of 6.1 and 5.2 than in mineral soils with pH values of 6.8 and 8.0, and was slightly more rapid in the more acidic organic soil. Schoen and Winterlin (1987) conducted an extensive study of the effects of various soil factors and organic amendments on degradation of diazinon. The factors affecting the rate of diazinon degradation in soil were pH, soil type, organic amendments, soil moisture, and pesticide concentration. Soil pH was a major factor affecting degradation. At a soil concentration of 100 ppm diazinon and 50% water saturation, estimated half-life values (first-order kinetics) at pH 4, 7, and 10 were 66, 209, and 153 days, respectively, in sandy loam; 49, 124, and 90 days, respectively, in clay loam; and 14, 45, and 64 days, respectively, in sandy loam amended with peat. Loss of diazinon occurred in the order of sandy loam with peat > clay loam > sandy loam. Addition of acidic peat to the soil lowered the pH and could have been responsible for increased hydrolysis. Degradation of diazinon in soil was most favorable when the pesticide was present at low concentrations in moist soil, amended with peat or acidified to a pH of 4, and least favorable at high diazinon concentrations in neutral or basic mineral soil.

In six types of soils, Somasundaram et al. (1991) reported that diazinon was hydrolyzed to 2-isopropyl-6-methyl-4-hydroxypyrimidine and that the degradation product was significantly more mobile in these soils than its parent compound diazinon. In an earlier study, Somasundaram et al. (1989) found that prior applications of 2-isopropyl-6-methyl-4-hydroxypyrimidine did not. enhance degradation of diazinon.

In a study of degradation of diazinon in three submerged tropical soils, only 2-6% of the originally applied diazinon remained 50-70 days postapplication (Sethunathan and MacRae 1969). Degradation of diazinon was more rapid in nonsterilized soils, indicating microbial participation in two of the three

soil types. In the third type (an acid clay soil), diazinon degradation was more rapid in the sterilized samples at pH 4.7, apparently because of the compound's instability under acid conditions. *Streptomyces sp*. isolated from the submerged soils could degrade the diazinon. In a field study of a treated cranberry bog by Szeto et al. (1990), disappearance of diazinon from irrigation ditch sediment (pH 4.4) and from sediment in an adjacent reservoir (pH 5.0) was equally rapid. These authors found that less than 1% of diazinon remained 38 and 22 days postapplication in the irrigation ditch and reservoir sediments, respectively. In nonsterilized soil, diazinon degradation was faster at 100% water saturation than at 50% water saturation. These results suggest that microbial activity under anaerobic conditions plays an important role in diazinon degradation (Schoen and Winterlin 1987).

Photolysis of diazinon on soil surfaces was studied by Burkhard and Guth (1979). The effectiveness of photolysis in 24 hours was only slightly greater on moist soil surfaces (51%) than it was on dry soil surfaces (44%) at 45 °C. The major photolytic product identified for diazinon was 2-isopropyl-6-methyl-4-hydroxypyrimidine. This same reaction product was found for acid hydrolysis and photolysis in aqueous solutions or on soil.

Gunner and Zuckerman (1968) reported synergistic microbial degradation of diazinon by two microorganisms, *Arthrobacter sp*. and *Streptomyces sp*. When *Arthrobacter sp*. and *Streptomyces sp*. were incubated separately on growth media where diazinon was the primary carbon source, neither was able to convert the pyrimidinyl carbon to carbon dioxide. When incubated together, only 6% of the parent diazinon remained, and 94% was converted to two unidentified metabolites. Two microorganisms isolated from flood soils also were found to hydrolyze diazinon (Adhya et al. 1981). Diazinon was rapidly hydrolyzed within 24 hours by both *Flavobacterium sp*. and *Pseudomonas sp*. A hydrolysis product of diazinon, 2-isopropyl-6-methyl-4-hydroxypyrimidine, was metabolized more rapidly by the *Flavobacterium sp*. than the *Pseudomonas sp*. More recently, oxypyrimidine was reported to be the major soil degradation product of diazinon and is considered to be more persistent than diazinon (EPA 1990b). Barik and Munnecke (1982) reported that an enzyme (parathion hydrolase) obtained from *Pseudomonas sp*. cultures could hydrolyze diazinon in soils. More than 98% of 10,000 ppm of diazinon in soil can be degraded within 24 hours if sufficient buffer and enzyme are added to the contaminated soil. The authors report that it is technically feasible to use parathion hydrolase to clean up diazinon spills in the environment.

Levanon et al. (1994) studied the effects of plow tillage on microbial activity and the degradation of diazinon in the O-5 cm soil layer. In no-tillage soils, higher microbial populations and activity were associated with higher mineralization rates of diazinon (45% mineralization after 76 days). Enhanced transformation rates played a role in minimizing leaching from no-tillage soils. Synergistic effects between fungi and bacteria in the degradation of diazinon were also observed. The authors noted that almost no mineralization of the compound occurred when either fungi or bacteria were selectively inhibited, demonstrating synergism between the two microbial communities. A higher proportion of diazinon leached from the plow tillage soils than from the no-tillage soils. Microbial population and activity measured as biomass, bacterial counts, hyphal length of fungi, and carbon dioxide evolution were all higher in samples of no-tillage soils.

5.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

Most information on diazinon concentrations in various environmental media derived from large scale monitoring networks dates from before the mid-1980s and no longer reflects current conditions. There is a noticeable lack of national monitoring studies that would allow meaningful estimation of current diazinon concentrations associated with various environmental media. Reliable evaluation of the potential for human exposure to diazinon depends in part on the reliability of supporting analytical data from environmental samples and biological specimens. In reviewing data on diazinon levels monitored in the environment, it should also be noted that the amount of chemical identified analytically is not necessarily equivalent to the amount that is bioavailable.

5.4.1 Air

Diazinon concentrations in the atmosphere were monitored in several national studies during the 1970s and 1980s and more recently in several regional studies. Diazinon has been measured in outdoor air samples in both rural and urban environments, near production facilities, and in indoor air (associated with its use for pest control in domestic and commercial buildings).

In a study of pesticide residues in ambient air sampled in 14-16 states during 1970, 1971, and 1972, diazinon was detected in 50% of the 2,479 samples analyzed, with a mean concentration of 2.5 ng/m^3 and a maximum concentration of 62.2 ng/m^3 (Kutz et al. 1976). Carey and Kutz (1985) reported that

ambient air concentrations of diazinon collected from February through September 1980 in Perkin, Illinois, ranged from 1.3 to 10 ng/m^3.

In a study of pesticide levels in ambient suburban air, diazinon was detected in 80, 80, and 40% of samples collected in three cities (Miami, Florida; Jackson, Mississippi; and Fort Collins, Colorado), respectively. The maximum diazinon concentration detected in each city was 3.9, 2.0, and 2.2 ng/m^3 for Miami, Florida; Jackson, Mississippi; and Fort Collins, Colorado, respectively (Kutz et al. 1976). During 1973-1974, diazinon concentrations in air were measured in urban Miami, Florida, and in the adjacent Everglades National Park. Urban diazinon levels ranged from not detectable to 3.3 ng/m^3 (1.5 ng/m^3 mean); corresponding levels in Everglades National Park ranged from not detectable to 1.9 ng/m^3 (0.6 ng/m^3 mean) (Lewis and Lee 1976). Nationwide, diazinon was detected in 48% of 123 urban air samples collected in ten U.S. cities during 1980. The maximum diazinon concentration reported was 23 ng/m^3 (mean 2.1 ng/m^3) (Carey and Kutz 1985).

Most recently, non-occupational exposure to diazinon among residents of two U.S. cities (Jacksonville, Florida, and Springfield, Massachusetts) was studied over three seasons: summer 1986, spring 1987, and winter 1988 (Whitmore et al. 1994). The study focused primarily on inhalation exposures with primary environmental monitoring consisting of 24-hour indoor and outdoor air. For the Jacksonville, Florida, population, the estimated mean diazinon concentrations ranged from 85.7 to 420.7 ng/m^3 in indoor air and 1.1 to 13.8 ng/m^3 in outdoor air. For the Springfield, Massachusetts, population, mean exposures were much less. The estimated diazinon concentrations ranged from 2.5 to 48.4 ng/m^3 in indoor air and 8.2 to 9.2 ng/m^3 in outdoor air.

Ambient diazinon concentrations were measured under foggy atmospheric conditions in and around the Central Valley of California (Parlier, California), which is a prime agricultural area dominated by fruit, nut, and citrus orchards (Glotfelty et al. 1990a; Seiber et al. 1993; Zabik and Seiber 1993). In fog, diazinon concentration was 1.6 ng/m^3 and diazoxon (the oxon transformation product) concentration was 0.82 ng/m^3- In a similar study, Schomburg et al. (1991) analyzed air and fog near Monterey, California, to determine whether the uptake of diazinon in advected oceanic fog was different from uptake in fog collected under stagnant inversion conditions in the Central Valley of California. Fog water concentrations of diazinon ranged from 0.15 to 4.8 µg/L (ppb) in coastal areas; higher concentrations ranging from 0.3 1 to 18 µg/L (ppb) were found in the Central Valley area. Diazinon and diazoxon favored the aqueous phase in foggy atmosphere, with 62.4 and 87.8%, respectively,

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reported in the aqueous phase. Zabik and Seiber (1993) studied the atmospheric transport of diazinon from California's Central Valley to the Sierra Nevada Mountains. Air samples collected from January through February 1991 represented the simultaneous collection of both vapor and particulate phases. Concentrations of diazinon and its oxon ranged from 13 to 10,000 pg/m^3 (0.013-10 ng/m^3) and 4 to 3,000 pg/m^3 (0.004-3 ng/m^3), respectively, for samples collected at the 114 m elevation and from 1.4 to 12 pg/m^3 (0.0014-0.012 ng/m^3) and 1.8 to 13 pg/m^3 (0.0018-0.013 ng/m^3), respectively, at the 533 m elevation. The pesticide concentrations in air samples decreased with distance and elevation moving east from the Central Valley into the higher elevations of the Sierra Nevada Mountains. At times, air concentrations of diazinon at the 114 m elevation were 1,000 times greater than concentrations detected at 533 m elevation. Concentrations at the 1,920 m elevation were typically below the limit of quantification. Wet deposition samples (rain and snow) collected at the 114 m elevation contained up to 6,100 pg/mL (6.1 ppb) diazinon and 2,300 pg/mL (2.3 ppb) diazoxon.

Diazinon residues in ambient air sampled within 800 m of two pesticide formulation plants in Arkansas (from 1970 to 1972) and within 275 m of a pesticide formulation plant in Tennessee (in 1971) ranged from 0.3-18.0 ng/m^3 (mean 2.2 ng/m^3) and 0.5-27.9 ng/m^3 (mean 7.3 ng/m^3), respectively (Lewis and Lee 1976).

In addition to its presence in the ambient atmosphere, diazinon also has been monitored in both outdoor and indoor air associated with its use in a variety of domestic, commercial, and occupational exposure situations. Exposure to diazinon from its use in lawn and home garden applications was evaluated by Davis et al. (1983). Diazinon was mixed with water and sprayed using compressed air sprayers or hose-end sprayers, and potential respiratory and dermal exposures were estimated from residues collected from respirator filters, body pads, and hand rinsings. These authors reported mean respiratory exposures of 1.9, 2.9, and 7.4 µg/hour associated with use of compressed air sprayers on lawns, compressed air sprayers on shrubs, and hose-end sprayers on lawns, respectively. The amount of diazinon collected in the respiratory pads was negligible compared to the amount collected on dermal pads. Total dermal exposures were 5,700, 7,500, and 29,000 pg/hour, respectively, for the three sprayer types; however, dermal exposure of the hands alone accounted for ≥85% of the total dermal exposure for each sprayer type..

Diazinon air concentrations related to vapors released from pest control strips were measured by Jackson and Lewis (1981). Diazinon levels in indoor air increased from 0.32 µg/m^3 at 6 hours after

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application of the pest strips to 1.34 $\mu g/m^3$ on day 15, and then declined to 1.21 $\mu g/m^3$ on day 30. Air sampling in a retail garden store where pesticide containers with diazinon were displayed showed an average diazinon concentration of 3.4 $\mu g/m^3$ (Wachs et al. 1983).

Currie et al. (1990) evaluated the concentrations of diazinon in indoor air and on working surfaces for a period of 10 days after application in commercial offices. The highest concentrations of diazinon (163 and 158 yg/m^3) were measured 4 hours postapplication in two empty offices, while the concentration in the furnished office was 27 $\mu g/m^3$. One day postapplication, levels were 125 and 70 $\mu g/m^3$ in the two empty offices and 27 $\mu g/m^3$ in the furnished office. Air concentrations of diazinon continued to decline and on day 6 postapplication were approximately 3.5 $\mu g/m^3$ in the empty offices and 8 $\mu g/m^3$ in the furnished office. Airborne levels of diazinon were distinctly lower in the furnished office, and this was attributed to obstruction of the applicator's spraying path by office furniture so that a lower amount of diazinon was applied. Diazinon deposition on aluminum plates was measured as an indicator of surface contamination, measurements ranged from 0.4-15 ng/cm^2. No overall decrease in surface contamination occurred over time. Plates suspended 1.5-2.1 m above the floor generally exhibited higher diazinon levels 24 hours post-treatment than at l-2 hours posttreatment. The authors believe this was a result of evaporation of diazinon from the carpeted floor augmented by air turbulence. Diazinon contamination measured by surface wipes on furniture and foil on carpet ranged from 13 to 38 ng/cm^2.

Diazinon levels in indoor air were monitored in an animal facility treated monthly with a 1% aqueous diazinon formulation (Williams et al. 1987). Indoor air sampling was conducted in two areas frequented regularly by facility personnel, the lounge and cage-washing areas. The lounge areas were enclosed rooms while the cage-washing areas were open-ended and were in effect part of the corridor system of the facility. Air samples were collected using adsorbent sampling tubes (Supelco-20 P) for 4 hours at 1.8 L/minute just prior to spraying on days 0, 28, and 56, approximately 16-20 hours posttreatment, and at various intervals thereafter. Diazinon levels increased immediately after spraying, but decreased rapidly to.2-3 $\mu g/m^3$ in less than 1 day and continued to decrease to less than 0.05 $\mu g/m^3$ until the next spraying. During many months of diazinon application there was little buildup in background diazinon air levels (<0.5 $\mu g/m^3$).

Lenhart and Kawamoto (1994) reported air concentrations of up to 297 $\mu g/m^3$ in greenhouse air after spray applications of an emulsifiable concentrate of diazinon (Clean Crop AG500), and concentrations up to 3,030 $\mu g/m^3$ in greenhouse air after a 4-hour cold fogging application of the same formulation.

Palmgren and Lee (1984) collected samples of grain dust (dust accumulated in the dust collection systems of grain elevators) from six grain elevators located in the New Orleans, Louisiana, area to evaluate potential occupational exposures of grain elevator personnel. Diazinon concentrations in grain dust were <.0.01 $\mu g/g$ for all 31 samples collected. The authors concluded that the concentration of diazinon on the grain dust posed no hazard.

5.4.2 Water

Since diazinon is not a priority pollutant and has not been considered to pose serious threats from bioconcentration or bioaccumulation in fish and shellfish species, it has attracted far less attention in the United States than persistent organochlorines like DDT or chlordane in routine surface water monitoring networks. Carey and Kutz (1985) reported that the maximum diazinon residue collected in a national surface water monitoring program conducted from 1976 to 1980 was 2.38 ppb and that diazinon was detected in only 1.2% of the samples collected. More recently, Pereira and Hostettler (1993) conducted a study of the Mississippi River and its tributaries during 1991 and 1992. These authors reported that diazinon was detected in water samples from the Illinois River at concentrations of 20 ng/L (0.02 ppb) and from several sites on the mainstem of the lower Mississippi River at concentrations ranging from 4 to 10 ng/L (0.004-0.010 ppb). During 1991, Domagalski and Kuivila (1993) monitored diazinon concentrations in water and suspended sediment collected at various sites in San Francisco Bay during low river discharge and after spring rain events. Diazinon was detected in water only after the spring rains and most (98%) of the diazinon was in the dissolved phase. Concentrations dissolved in the water column ranged from 4.6 to 14.6 ng/L (0.005-0.015 ppb). The authors suggest that diazinon may be close to equilibrium with respect to sorption or desorption on suspended sediment particles.

In the Great Lakes region, diazinon was detected in surface waters in several river basins in southern Ontario, Canada. Braun and Frank (1980) monitored surface water concentrations of 8 organochlorine and 12 organophosphate pesticides in 11 agricultural watersheds in southern Ontario. All watersheds drained into the Great Lakes. Diazinon residues as a result of field use were detected in only one

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water shed, but the chemical was repeatedly detected in 34% of samples (1975-76) and 74% of
samples (1976-77) collected from one creek. The source of the diazinon was traced to its indoor use
to control flies in a series of mushroom houses that discharged via a drainage tile system directly to
the creek. The maximum residues of diazinon in the stream were 140 ppb (5.75 ppb mean) and
26 ppb (1.02 ppb mean) in 1975-76 and 1976-77, respectively. In a more recent study, Frank and
Logan (1988) measured pesticide and industrial chemical residues at the mouth of the Grand, Saugeen,
and Thames Rivers in southern Ontario, Canada, from 1981 through 1985. River water samples
collected at the mouths of the three rivers (that drain into the Great Lakes) were analyzed for
20 herbicides, 3 fungicides, and 25 insecticides including diazinon. One water sample collected during
May through August 1982 contained a mean diazinon concentration of 0.21 ppb. Maguire and Tkacz
(1993) monitored concentration of pesticides in surface water near the mouths of the Yamaska River in
Quebec, Canada, and five of its tributaries during 1986 and 1987. Diazinon was detected at the mouth
of the Yamaska River at concentrations ranging from 2.1 to 11.9 ng/L (0.002-0.012 ppb), at the mouth
of the Saint-Nazaraire River at concentrations ranging from 3.1 to 26.7 ng/L (0.003-0.027 ppb), and at
the mouth the Salvail River at concentrations ranging from 1.1 to 4.9 ng/L (0.001-0.005 ppb).
Frank et al. (1990a) conducted a survey of 211 rural ponds in southern Ontario and measured
concentrations of 29 herbicides, fungicides, and insecticides including diazinon. Two ponds were
found to be contaminated with diazinon, and residues in pond water ranged from 0.6 to 1.7 ppb (1.2 ±
0.8 ppb mean). The source of the diazinon in these two cases was attributed to accidental pesticide
spills during agricultural application.

Diazinon concentrations in water have also been monitored in the United States and in several
Canadian studies associated with the use of the compound in agricultural applications. Kendall et al.
(1993) monitored diazinon residues in ponds and creeks adjacent to a golf course in coastal
Washington where two turf applications of diazinon were made at a rate of 2.2 kg active ingredient
per hectare. A maximum diazinon residue of 17 ppb was measured in the study area ponds and
creeks. Wan et al. (1994) monitored concentrations of diazinon and six other organophosphate
pesticides in farm ditches of the lower Fraser River Valley of British Columbia, Canada, from July to
December 1991. These authors reported that diazinon was consistently found in ditch water (81% of
samples) at 7 locations with a mean concentration of 0.07 μg/L (ppb) (range of 0.01-0.34 μg/L [ppb]).
The percentage of positive detections for diazinon in water samples was 81%. The presence of
diazinon in ditch water was correlated with consistent detection of diazinon residues in soils from
nearby fields. Szeto et al. (1990) monitored the persistence of diazinon in coastal cranberry bogs and

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adjacent surface waters in British Columbia, Canada. Bogs were treated with two applications of diazinon 5G (granules) at a rate of 6 kg active ingredient per hectare approximately 2 weeks apart. One day after the first and second applications, maximum concentrations of diazinon in water in an irrigation ditch were 338 ppb and 456 ppb, respectively. Maximum concentrations in an adjacent reservoir were 78.5 ppb and 58.1 ppb for the first and second treatments, respectively. Water samples collected immediately outside the diked bog area contained a maximum of 29.1 ppb diazinon, but concentrations were usually <10 ppb. Tributary water 100 m downstream from the cranberry bog site contained a maximum diazinon residue of 2.8 ppb.

Recently, acute toxicity of sewage treatment plant effluents to aquatic bioassay testing organisms in the United States has been tied to diazinon (Amato et al. 1992; Burkhard and Jenson 1993). Given the considerable use of diazinon in urban areas, diazinon in sewage treatment effluents is not unexpected. Urban nonpoint source inputs from diazinon-impregnated yard wastes, runoff from treated lawn and garden areas, or illegal dumping may require increased pollution prevention efforts through the National Pollution Discharge Elimination System (NPDES) program in many larger cities (Amato et al. 1992; Burkhard and Jenson 1993). A maximum diazinon residue of 1.7 ppb in POTW effluents was associated with the toxic fraction in effluent bioassay tests with *Ceriodaphnia dubia* (Burkhard and Jenson 1993). Amato et al. (1992) suggest that the significance of detecting diazinon at acutely toxic concentrations in municipal waste water may indicate a more widespread problem.

In a groundwater contamination study of 28 of California's 58 counties that evaluated over 50 pesticides (from both point and nonpoint sources), diazinon was detected in 12 samples (Cohen 1986). Diazinon is included as an analyte of interest in the EPA Pesticides in Ground Water Database (EPA 1989a) and was detected at two sites. A detection in California was related to point source contamination (residue level was unspecified), and a detection of 478 ppb (maximum) and 162 ppb (mean) in Mississippi was in an area where appreciable agricultural use of pesticides occurs. In the Great Lakes region, diazinon was found in a survey of rural wells in southern Ontario, Canada, monitored between 1979 and 1984 (Frank et al. 1987) and in farm wells monitored between 1986 and 1987 (Frank et al. 1990b). However, no concentrations of diazinon in groundwater were provided by these authors.

5.4.3 Sediment and Soil

Diazinon has not been the focus of many national soil or sediment monitoring programs in the United
State, but has been monitored in regional studies associated with agricultural applications in both the United
States and Canada. In a national surface water quality monitoring study (1976-1980), diazinon was detected in
0.5% of the sediment samples analyzed, with a maximum residue of 7.1 ppb (Carey and Kutz 1985).
Domagalski and Kiuvila (1993) reported concentrations of diazinon in suspended sediments from various sites
from San Francisco Bay ranging from not detected to 2.8 ng/g (ppb).

Soil contamination of diazinon ranging from 95.5 mg/m^2 (2 hours postapplication) to 35.6 mg/m^2 (342 hours
postapplication) resulted from spray applications of 4.5 kg diazinon (50 WP formulation) per hectare to a
dormant peach orchard in the Central Valley of California (Glotfelty et al. 1990b). Diazinon concentrations in
sediments of a cranberry bog treated with two applications of diazinon (Diazinon 5G at 6 kg active ingredient
per hectare) were measured by Szeto et al. (1990). These authors reported that the highest diazinon residues
were 21 ppm (21,000 ppb) (wet weight) in sediments of irrigation ditches collected 4 days postapplication. The
maximum sediment concentration measured in an adjacent reservoir was 2 ppm (2,000 ppb). Four days
postapplication, the maximum sediment concentration was 80 ppb in a waterway outside the diked bog and
only 10 ppb in a tributary 100 meters downstream from the bog. Wan et al. (1994) monitored ditch water,
soils, and sediments from July to December 1991 in an agricultural area in the lower Fraser River Valley of
British Columbia, Canada. Diazinon concentrations in ditch sediment were detected at three sites; the mean
concentrations were 8, 2, and 38 µg/kg (ppb) at the Vancouver, Cloverdale, and Sumas Prairie sites,
respectively. Diazinon was also detected in topsoil (<5 cm deep) at five sites; the mean concentrations
were 268 µg/kg (268 ppb) (range of 2-3,307 µg/kg), 5 µ/kg (ppb) (range of l-9 µg/kg), 769 µg/kg
(ppb) (range of 13-2862 µg/kg), 13 µg/kg (ppb) (range of 4-30 µg/kg) , and 39 µg/kg (ppb) (range of
l-236 µg/kg) at the Westham Island, Ladner, Bumaby, Cloverdale, and Sumas Prairie sites,
respectively. The concentrations at all these stations declined from July to December.

5.4.4 Other Environmental Media

Braun and Frank (1980) reported diazinon residues in three fish species collected from a creek in
southern Ontario, Canada, contaminated from a point source discharge. Tissue residues for the three
edible fish species were 18 ppb in the brown bullhead *(Ictalurus nebulosus),* 17 ppb in the black
crappie (*Pomoxis nigromaculatus*), and 92 ppb in the gizzard shad (*Dorosoma cepedianum*). The

maximum diazinon concentrations measured in the contaminated creek water for 1975-76 and 1976-77 were 140 ppb (5.75 ppb mean) and 26 ppb (1.02 ppb mean), respectively.

Pesticide residue data in domestic and imported foods and animal feeds from 1982 to 1986 were evaluated by Hundley et al. (1988). These authors reported that diazinon was detected in a wide variety of domestic foods: garbanzo beans, green beans, pinkos dry beans, broccoli, celery, limes, collards, cucumbers, endives, butter lettuce, green leaf lettuce, iceberg lettuce, watermelons, green onions, parsley, Chinese peas, spinach, Italian squash, Swiss chard, and tomatoes. None of the samples exceeded EPA tolerance limits. Diazinon was also detected in the following imported foods: apples, broccoli, dried cherries, cucumbers, feijoa, kiwi fruit, green leaf lettuce, red leaf lettuce, romaine lettuce, cantaloupe, okra, green onions, Chinese peas, Anaheim peppers, bell peppers, caribe peppers, jalapeno peppers, serano peppers, prunes, raisins, Italian squash, yellow squash, and tomatoes. Only residues in caribe peppers exceeded the EPA tolerance limit. Concentrations of diazinon in ready-to-eat foods were monitored for 10 years from 1982 to 1991 through the FDA's Revised Market Basket Survey (KAN-DO Office and Pesticide Teams 1995). Diazinon was detected in 894 samples of 144 different foods at a mean concentration of 0.0019 μg/g (1.9 ppb).

The frequency of detection of diazinon in the FDA Total Diet Study conducted from 1982 to 1984 was 13% (Gunderson 1988). Diazinon intakes, in μg/kg body weight/day, estimated for these total diet analyses (1982-1984) were 0.0121, 0.0129, and 0.0073 for 6-l l-month-old infants, 14-16-year-old males, and 60-65-year-old females, respectively. More recently, the frequency of occurrence of diazinon detections in the FDA Total Diet Study declined to 9% in 1989 (FDA 1990), 6% in 1990 (FDA 1991), 4% in 1991 (FDA 1992), 5% from 1991 to 1993 (FDA 1994), and 5% in 1994 (FDA 1995). Diazinon intakes in μg/kg body weight/day, estimated for the total diet analyses also declined from intakes estimated in the 1982-84 analysis and were 0.0031, 0.0034, and 0.0017 in 1989 (FDA 1990); 0.0026, 0.0022, and 0.0017 in 1990 (FDA 1991); and 0.0049, 0.0022, and 0.0022 in 1991 (FDA 1992) for 6-l l-month-old infants, 14-16-year-old-males, and 60-65-year-old-females, respectively.

5.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

While no quantitative information is available on the percentage of diazinon released to each environmental compartment, diazinon can be emitted to any or all environmental media (air, surface

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water, groundwater, and soil) depending on the source of the release, formulation used, and prevailing environmental conditions. General population exposure to diazinon may occur through three routes: dermal contact, inhalation, and ingestion of contaminated food or drinking water. The major routes of exposure to diazinon for the general population are through dermal contact directly with the chemical during domestic application for control of home and garden pests; through dermal contact with diazinon-treated plant materials such as grass clippings; or through dermal contact with treated surfaces (e.g., furniture) in domestic or office buildings. For children particularly, a potential source of exposure can be related to the indoor application of diazinon on furniture, rugs, and flooring. The general population may also be exposed to diazinon through inhalation of contaminated ambient (outdoor) air particularly in agricultural areas where diazinon is extensively used or in urban areas where it is applied to lawns and gardens. Since many commercial buildings and residential buildings are sprayed with diazinon or use pest control strips that vaporize diazinon, there is the possibility of exposure from inhalation of vapors in these diazinon-treated indoor air spaces. The oral route of exposure may include ingestion of foods contaminated with small residues of diazinon or consumption of contaminated drinking water.

Davis et al. (1983) reported that dermal exposure to diazinon from spray applications of the compound for home and garden applications ranged from 5,700 to 29,000 μg/hour depending on the type of sprayer used. The mean respiratory exposures ranging from 1.9 to 7.4 μg/hour, were negligible compared to the dermal exposures. In addition, these authors reported that dermal exposure of the hands, which accounted for 85% or more of the total dermal exposure, could be easily reduced by the use of protective gloves.

Non-occupational exposure to diazinon for residents of two U.S. cities (Jacksonville, Florida, and Springfield, Massachusetts) were studied over three seasons: summer 1986, spring 1987, and winter 1988 (Whitmore et al. 1994). The study focused primarily on inhalation exposures with primary environmental monitoring consisting of 24-hour indoor air, personal air, and outdoor air. For the population of Jacksonville, Florida, the mean diazinon concentration ranges were 85.7-42.0.7 ng/m^3 for indoor air, 1.1-13.8 ng/m^3 for outdoor air, and 89.0-321.6 ng/m^3 for personal air. For the population in Springfield, Massachusetts, mean exposures were much less. The diazinon concentrations were 2.548.4 ng/m^3 for indoor air, 8.2-9.2 ng/m^3 for outdoor air, and 1.4-10.1 ng/m^3 for personal air. The mean air exposure for diazinon in Jacksonville, Florida, was 1,380 ng/day, and dietary exposures were 590-1,140 ng/day. The mean air exposure estimated for Springfield, Massachusetts, was almost

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10 times lower (158 ng/day), while the dietary exposure (586 ng/day) was equal to the low end of the range for the population of Jacksonville, Florida. In Jacksonville, Florida, characterized as a high pesticide use area, inhalation exposure exceeded dietary exposure; in Springfield, Massachusetts, characterized as a low pesticide use area, the dietary exposure to diazinon exceeded the inhalation exposure.

Workers employed in industries that manufacture, formulate, package, or apply diazinon and workers involved in the disposal of diazinon or diazinon-containing wastes have the potential to be exposed to the highest concentrations of diazinon. In occupational settings, dermal exposure and subsequent absorption through intact skin is the most important route of exposure, and inhalation exposure is generally less important (Jeyaratnam and Maroni 1994). Inhalation of diazinon depends on its volatility, the type of formulation used, and the application technique employed. Occupational ingestion may occur as a result of poor work practices and/or lack of personal hygiene.

NIOSH recommends that the occupational exposure level not exceed 100 $\mu g/m^3$ for a 10-hour TWA workday (NIOSH 1992). In addition, the American Conference of Governmental Industrial Hygienists has recommended a time-weighted average threshold limit value (TWA-TLV) of 100 $\mu g/m^3$ for occupational exposure to diazinon (ACGIH 1986).

Except for professional pesticide applicators or farm workers, the exposure risks from diazinon appear relatively minor as long as label instructions are followed and safeguards are taken to avoid extensive dermal contact. Even studies of dermal exposure typical of shearers handling sheep that have been dipped in diazinon showed dermal absorption rates of less than 4% (Wester et al. 1993). Studies of dermal exposure for workers in grain elevators failed to detect diazinon in grain dust above the 0.01 $\mu g/g$ detection limit, although much higher levels have been reported from Australia (Palmgren and Lee 1984).

The use of a 2-day lag period from the time of diazinon application to the use of office or domestic indoor space appears adequate to eliminate exposure risks from vapors and residues that might be incurred from either inhalation or dermal absorption. Air sampling of a room treated with 36 pest control strips measured a maximum diazinon air concentration of 1.34 $\mu g/m^3$ 15 days postapplication (Jackson and Lewis 1981). Similarly, Williams et al. (1987) found that air sampling in two animal facility areas used by facility personnel and treated monthly with a 1% aqueous diazinon solution

measured 2-3 μg/m^3 less than 24 hours postapplication. Currie et al. (1990) also measured diazinon air concentrations in empty and furnished offices treated with a 1% aqueous solution. Four hours postapplication, diazinon air concentrations were 163 and 158 μg/m^3 in two empty offices and 28 μg/m^3 in the furnished office. One day postapplication, diazinon levels in the offices ranged from 125 μg/m^3 (empty office) to 27 μg/m^3 (furnished office), but by 2 days postapplication the highest diazinon air concentration measured was 53 μg/m^3. Air sampling levels of diazinon 2 days posttreatment in these three indoor exposure contexts were well below the NIOSH &hour TWA permissible exposure level (PEL) of 100 μg/m^3.

Residual air concentrations of diazinon in a commercial greenhouse were studied by Lenhart and Kawamoto (1994). These authors monitored diazinon air concentrations applied as a spray and by cold fogging. The 40 minute spray application was made to a portion of the greenhouse with only passive ventilation (adjustable window vents). During application, circulating fans were turned off and all roof vents were closed. After the spray application, 1.4 L of the diazinon emulsifiable concentrate formulation in 18 L of water were added to each of two cold fogging machines set for a 4-hour cold fogging application. Air samples were collected during the work shift prior to pesticide application, hourly during the application, and for 4 consecutive days after the pesticide application. Full shift area air samples were collected. During the postapplication period, air circulating fans were continuously operated and the roof vents were open occasionally. The 8-hour TWA for the spray application ranged from not detected to 25 μg/m^3. The 8 hour TWA diazinon concentrations ranged from 6.0 to 52 μg/m^3 (Saturday), 3 to 30 μg/m^3 (Sunday), 2.4 to 17 μg/m^3 (Monday), and not detected to 12 μg/m^3 (Tuesday). During the cold fogging application, diazinon concentrations on Friday ranged from 730 to 3,030 μg/m^3. Residual 8-hour TWA concentrations for this application ranged from 70 to 250 μg/m^3 (Saturday), 27 to 67 μg/m^3 (Sunday), 20 to 59 μg/m^3 (Monday), and 19 to 40 μg/m^3 (Tuesday). Two of the 4 samples collected on Saturday exceeded the NIOSH TWA permissible exposure level of 100 μg/m^3 for occupational exposures to diazinon. Results of this study indicate that greenhouse workers can be at risk of inhalation exposure to residual diazinon concentrations. The authors believe that all diazinon applications should be conducted on Friday evenings after the greenhouse workers have left so that much of the residual pesticide can settle over the weekend.

Finally, air sampling at a retail garden store conducted to determine exposures for retail employees showed levels of diazinon averaging only 3.4 μg/m^3, well below the NIOSH TWA exposure level of 100 μg/m^3 (Wachs et al. 1983). However, these authors point out that the air concentrations they

reported may vary greatly among retail stores depending on the amounts and types of diazinon formulation sold, air temperature, condition of the packaging material (e.g., tom packaging, loose lids), prior spills, and types of floor coverings.

The National Occupational Exposure Survey (NOES) conducted by NIOSH from 1981 to 1983 estimated that 39,342 workers (including 3,216 women) employed at 3,168 facilities were potentially exposed to diazinon in the United States (NOES 1990). The NOES database does not contain information on the frequency, concentration, or duration of exposure; the survey provides only estimates of workers potentially exposed to chemicals in the workplace.

5.6 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Other than individuals who are occupationally exposed to diazinon (during its production, formulation, packaging, distribution, use, or disposal), populations exposed to higher than background concentrations of diazinon in ambient air include those living near chemical manufacturing or processing sites, individuals living on farms or in the vicinity of agricultural areas where diazinon is extensively used, and individuals living near hazardous waste sites. Individuals living near these sites may also be exposed to potentially higher concentrations of diazinon or its metabolites in their drinking water if they obtain tap water from wells located near these sources. Children may receive higher diazinon doses from dermal exposures if they play on freshly treated lawns or soil. In addition, children may receive potentially higher oral doses from ingestion of diazinon-treated soils from their hands while playing in contaminated areas.

5.7 ADEQUACY OF THE DATABASE

Section 104(i)(5) of CERCLA, as amended, directs the Administrator of ATSDR (in consultation with the Administrator of EPA and agencies and programs of the Public Health Service) to assess whether adequate information on the health effects of diazinon is available. Where adequate information is not available, ATSDR, in conjunction with the NTP, is required to assure the initiation of a program of research designed to determine the health effects (and techniques for developing methods to determine such health effects) of diazinon.

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The following categories of possible data needs have been identified by a joint team of scientists from ATSDR, NTP, and EPA. They are defined as substance-specific informational needs that if met would reduce the uncertainties of human health assessment. This definition should not be interpreted to mean that all data needs discussed in this section must be filled. In the future, the identified data needs will be evaluated and prioritized, and a substance-specific research agenda will be proposed.

5.7.1 Identification of Data Needs

Physical and Chemical Properties. While the principal properties of diazinon are well characterized, (ASTER 1995; Howard 1991; HSDB 1996; Merck 1989) there are data gaps for melting point, odor and taste thresholds, autoignition temperature, flash point, and explosive limits for the compound. Additional information on these properties would be helpful in assessing the compound's environmental fate. There are also data gaps for some spontaneously-produced degradation products some of which may be as toxic or more toxic than diazinon.

Production, Import/Export, Use, Release, and Disposal. As with many pesticide agents, limited current information was found on production, import and export volumes, or even on registered use patterns for diazinon. No information was available from the Toxics Release Inventory on facilities involved in the production or processing of diazinon because it was not one of the chemicals the facilities were required to report prior to January 1, 1995 (EPA 1995a, 1995b). This lack of information seriously compromises efforts to design monitoring programs to study fate and transport and can seriously jeopardize proper assessments of exposure opportunities and health risks for this compound.

According to the Emergency Planning and Community Right-to-Know Act of 1986, 42 U.S.C. Section 11023, industries are required to submit chemical release and off-site transfer information to the EPA. The Toxics Release Inventory (TRI), which contains this information for 1995, will become available in May of 1997. This database will be updated yearly and should provide a list of industrial - - production facilities and emissions.

Environmental Fate. Diazinon is moderately mobile in some soil types (Arienzo et al. 1994; Kenaga 1980; Sharom et al. 1980a). Information on the mobility of diazinon and on a major degradation product 2-isopropyl-6-methyl-4-hydroxypyrimidine in various soil types is available

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(Arienzo et al. 1994; Levanon et al. 1994; Sharom et al. 1980a; Somasundaram et al. 1991). In the atmosphere, diazinon is subject to degradation due to photolysis (Gore et al. 1971) and reactions with hydroxyl radicals (Glotfelty et al. 1990a; Schomberg et al. 1991; Seiber et al. 1993; SRC 1995). In water, diazinon is subject to hydrolysis, photolysis and biodegradation. The rate of degradation of diazinon in water and soil is strongly influenced by pH (Chapman and Cole 1982; Ferrando et al. 1992; Frank et al 1991b; Garcia-Repetto et al. 1994; Sharom et al. 1980b). Diazinon undergoes only slight photolysis in water, with reported half-life estimates ranging from 42 to 88 days (Frank et al. 1991b; Wolfe et al. 1976). Diazinon can be degraded at the soil surface by photolysis (Burkhard and Guth 1979), and in soils and sediment by hydrolysis (Chapman and Cole 1982; Levanon et al. 1994; Schoen and Winterlin 1987; Sethunathan and MacRae 1969; Somasundaram et al. 1989, 1991) and by biodegradation by microorganisms (Adhya et al. 1981; Batik and Munnecke 1982; Gunner and Zuckerman 1968). Additional information on the mechanism by which diazinon is converted to diazoxon in the atmosphere would be useful; additional information on the persistence and mobility of the major degradation products of diazinon would also be useful in evaluating the environmental fate of diazinon and its degradation products.

Bioavailability from Environmental Media. Diazinon can be absorbed following inhalation, dermal, or oral exposures. Absorption through the skin is of major concern for exposures of farmers, farm workers, commercial applicators, or homeowners related to the use of diazinon as an insecticide or nematocide (Davis et al. 1983). Absorption via inhalation is a major concern particularly with respect to indoor exposures to diazinon within 2 days postapplication of the compound as a pest control agent in commercial buildings and homes (Currie et al. 1990; Jackson and Lewis 1981; Lenhart and Kawamoto 1994; Williams et al. 1987). Additional information on the concentrations of diazinon in indoor air and in groundwater from domestic wells, particularly from environments near hazardous waste sites, is needed to determine the bioavailability of diazinon in these media.

Food Chain Bioaccumulation. Diazinon has an estimated low bioconcentration potential (BCF=77) (Kenaga 1980) in aquatic organisms, which is generally confirmed by measured BCF values obtained from laboratory studies with fish and other aquatic invertebrates (El Arab et al. 1990; Keizer et al. 1991; Sancho et al. 1993; Tsuda et al. 1989, 1995). Further information on measured BCF values for additional edible fish and shellfish would be helpful, as would information on tissue residues of diazinon and its major degradation products in edible species. No information was found on studies associated with plant uptake, but diazinon is rarely detected above EPA tolerance limits

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(Hundley et al. 1988). Bioaccumulation in aquatic food chains does not appear to be important, and no further information on biomagnification is required.

Exposure Levels in Environmental Media. Diazinon is distributed in all environmental media and has been detected in ambient air (Carey and Kutz 1985; Glotfelty et al. 1990a; Kutz et al. 1976; Lewis and Lee 1976; Schomburg et al. 1991; Seiber et al. 1993; Zabik and Seiber 1993), in indoor air (Currie et al. 1990; Jackson and Lewis 1981; Lenhart and Kawamoto 1994; Palmgren and Lee 1984; Wachs et al. 1983; Williams et al. 1987), surface water (Braun and Frank 1980; Carey and Kutz 1985; Domagalski and Kuivila 1993; Frank and Logan 1988; Frank et al. 1990a; Kendall et al. 1993; Maguire and Tkacz 1993; Pereira and Hostettler 1993; Szeto et al. 1990; Wan et al. 1994), groundwater (Cohen 1986; EPA 1989a), sediment (Carey and Kutz 1985; Domagalski and Kuivila 1993; Szeto et al. 1990), and some fish (Braun and Frank 1980). The levels of diazinon in air, surface water, groundwater, and soil have been well documented. There is a need for more information from national or large regional studies on current exposure levels. Additional information on tissue residues of diazinon and its major degradation products in edible fish and shellfish species would be particularly helpful in quantifying health risk from consumption of contaminated species.

Reliable monitoring data for the levels of diazinon in contaminated media at hazardous waste sites are needed so that the information obtained on levels of diazinon in the environment can be used in combination with the known body burden of diazinon to assess the potential risk of adverse health effects in populations living in the vicinity of hazardous waste sites.

Exposure Levels in Humans. Data regarding levels of diazinon in humans from environmental exposures (the general population, populations living near hazardous waste sites, or occupationally exposed groups) are not available. It is arguable that these levels are not knowable because of the rapid metabolism and clearance of diazinon after it enters the body (Iverson et al. 1975; Machin et al. 1975; Mount 1984; Mũcke et al. 1970). Additional studies which associate levels of diazinon in the environment and levels of diazinon metabolites in body tissues would be helpful. These studies are needed to give a practical assessment of exposure risks. This information is necessary for assessing the need to conduct health studies on these populations.

Exposure Registries. No exposure registries for diazinon were located. This substance is not currently one of the compounds for which a subregistry has been established in the National Exposure

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Registry. The substance will be considered in the future when chemical selection is made for subregistries to be established. The information that is amassed in the National Exposure Registry facilitates the epidemiological research needed to assess adverse health outcomes that may be related to exposure to this substance.

5.7.2 Ongoing Studies

The U.S. Department of Agriculture has sponsored several studies on diazinon.

Drs. G.K. Felton, W.W. Frye, and W. Witt at the University of Kentucky are studying the impact of agricultural systems on surface and groundwater quality. Research and management oriented catchment scale water quality models are being developed to describe the movement of pesticides and nutrients from soils into surface and groundwater. Two watersheds will be instrumented for continuous flow data collection and periodic water quality sampling (including data on diazinon). One watershed will be evaluated for chemical impacts on a mixed agriculture-forest-suburban watershed. The second watershed will be evaluated for chemical impacts due to shifts in land use.

The effects of floods on the removal of pesticide residues and nutrients from flooded areas used in cranberry production are being studied at the University of Massachusetts. Weekly water samples taken during the growing season will be analyzed to determine residues of diazinon, among other pesticides, near the flood gates of two cranberry bogs, in two conveying streams, and in three receiving waterbodies. Specimens of several edible shellfish species including blue crabs, scallops, and quahogs will also be analyzed for residues.

Dr. J.N. Seiber at the Range Wildlife and Forestry Department of the University of Nevada is studying the aerial transport and deposition of organophosphate insecticides including diazinon in Sierra Nevada forests. This study involves 1) developing methods for analyzing toxicants in air and atmospheric moisture; 2) determining concentrations and frequency of airborne residues of pesticides, conversion products, and other organic toxicants; 3) determining wet and dry deposition levels of toxicants in the Sierra Nevada and Great Basin areas; and 4) correlating the information with data on emissions, meteorology, physicochemical properties, and modeling. Pine needles from Ponderosa pines will be sampled to determine their ability to capture airborne pesticide residues.

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Dr. P. Jeffers at the State University of New York at Cortland is gathering information to determine the persistence of organophosphorus compounds in groundwater and the effects of various soils on the degradation and transport of these compounds. Both neutral and base hydrolysis processes will be . evaluated. Transport studies in soil columns will be conducted to determine the mobility of diazinon in soils.

At the Agricultural Research Service in Raleigh, North Carolina, investigations are underway by Dr. D.E. Moreland to 1) characterize and elucidate the oxidation of endogenous and exogeneous substrates including diazinon by plant cytochrome P-450 monooxygenases; and 2) identify treatments that stimulate the metabolism of diazinon and other pesticides, and lower pesticide residues in consumable agricultural plant products.

Dr. R.J. Wright at the Beltsville Agricultural Research Center, Beltsville, Maryland, is developing experimental approaches to allow direct measurement of pesticides including diazinon in wet and dry atmospheric deposition. The spatial and temporal distribution of airborne pesticide inputs to Chesapeake Bay will be determined to test predictive models of the atmospheric transport and deposition of agricultural chemicals.

Dr. A. Shaw at the University of Maryland Eastern Shore in Princess Anne, Maryland, is evaluating textile substrate for pesticide barrier effectiveness and comfort. Tests will be conducted to assess effectiveness of decontamination processes for these personal protection devices. Diazinon emulsifiable concentrates will be used to contaminate fabrics. Simulated wear studies will be conducted in the laboratory to assess the efficacy of these fabrics in protecting human health.

The U.S. Department of the Interior has also sponsored a study. Dr. C. Childress of the Department of the Interior, U.S. Geological Survey is conducting a surface water quality assessment for Region J, North Carolina. The purpose of this project is to gather long-term regional water-quality monitoring data for stream tributaries to the region's drinking water supplies. Many of these streams receive a complex combination of treated industrial and municipal effluents, in addition to nonpoint source urban and agricultural runoff. This project will supplement the existing state database for major ions, nutrients, and trace metals and create a new database on synthetic organic chemicals (e.g., diazinon). The purpose of this project is to 1) document spatial differences in regional surface water quality, 2) examine temporal trends in water quality, and 3) provide water quality data to local environmental

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planners and managers. The project will establish a fixed-interval monitoring network to assess water quality at 21 stream and 12 reservoir sites. Preliminary sampling has detected lindane, diazinon and/or dieldrin in nearly 50% of the water samples.

No additional information was located on current studies that would fill existing data needs for diazinon (FEDRIP 1995).